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Research Paper

Understanding avian assemblage change within anthropogenic environments using citizen science data

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ABSTRACT

Anthropogenic land use is a major driver of biodiversity loss, with different land use activities having a range of impacts on native communities. These myriad impacts make it difficult to identify the key drivers of species declines, especially across heterogenous anthropogenic environments. Our study aims to identify whether the species and traits being lost in disturbed environments differ across a land-use intensity gradient, in order to prioritise management effort in Greater Brisbane, Australia. We applied List Length Analysis (LLA) to standardise citizen-collected avian records, and model the change in prevalence for 182 bird species within urban, rural and forested environments. We then tested whether understorey-nesting, ground-nesting, insectivorous or small-bodied functional groups were significantly declining in prevalence within the entire avian assemblage. We found a greater probability of decline for small-bodied and understorey-nesting species in urban environments, lending support to established findings that, in urban environments of Greater Brisbane, competition with larger territorial birds and understorey loss are impacting communities. Our study also highlighted that the species declining and increasing in prevalence differed across the land use intensity gradient. Management approaches should therefore be targeted to mitigate the distinct impacts associated with particular land uses. In Greater Brisbane, managers should focus on maintaining urban understories and monitoring overabundant avian competitors. Where funds are limited, LLA represents a useful tool to harness non-standardised data, to guide early management and monitoring effort. Such tools equip managers to conserve biodiversity in anthropogenic environments.

1. Introduction

Population growth and development have led to rapid and ongoing urbanisation, transforming natural communities (Grimm et al., 2008; McKinney, 2008). Habitat clearing, introduction of non-native species, fragmentation, and various forms of pollution (sound, air, light, soil and water) interact to shape biotic communities, and have resulted in reduced assemblages of the regional species pool within cities (Aronson et al., 2014; Blair, 1996; Marzluff, 2001; Sol, Gonzalez-Lagos, Moreira, Maspons, & Lapiedra, 2014). These impacts however, are variable across the development gradient, thus resulting in distinct biotic communities forming within heterogeneous anthropogenic environments (Beninde, Veith, & Hochkirch, 2015; Blair, 1996; McKinney, 2008). Effective monitoring and analysis is therefore critical in order to a) differentiate between aspects of anthropogenic development and their associated impacts on biotic communities and b) manage those drivers having the greatest ecological impact (Aronson et al., 2014; Blair, 1996; Evans, Ryder, Reitsma, Hurlbert, & Marra, 2015; Lepczyk et al., 2017; Marzluff, 2016; Sol et al., 2014).

To distinguish between the multiple effects of land use change, studies both regionally and internationally have been examining the impacts of land use intensification on specific taxonomic and functional groups (Aronson et al., 2014; Chace & Walsh, 2006; Faeth, Bang, & Saari, 2011; Marzluff, 2016). An increase in the availability and volume of observational data on birds, has made trait analysis especially viable for avian taxonomic groups (Aronson et al., 2014; Chace & Walsh, 2006; Marzluff, 2016). A number of studies have identified that, in disturbed environments, birds with particular functional traits are either disappearing or beginning to dominate (Chace & Walsh, 2006; Croci, Butet, & Clergeau, 2008; Faeth et al., 2011; Kark, Iwaniuk, Schalimtzek, & Banker, 2007; Lepczyk et al., 2008; Lepczyk et al., 2017; van Rensburg, Peacock, & Robertson, 2009). In particular, a small

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subset of traits have demonstrated greater success in urban environments; with sociable, sedentary, long-lived, broadly-distributed, resource-flexible, phenotypically-plastic and species with small clutch-sizes having emerged as successful urban-exploiters (Aronson et al., 2014; Croci et al., 2008; Kark et al., 2007; Lepczyk et al., 2017; Marzluff, 2016). While these outcomes may suggest a homogenisation of species across cities, global-scale analyses have underscored that urban environments remain dominated by native species from the regional species pool (Aronson et al., 2014). Thus, understanding the processes occurring at a regional scale is important to manage biodiversity loss within anthropogenic environments (Aronson et al., 2014; Croci et al., 2008; Evans, Newson, & Gaston, 2009; Evans et al., 2015; Marzluff, 2016).

Within Australia, a few traits have repeatedly emerged as declining in disturbed environments. Understorey and ground-nesting traits are sensitive to development, due to these species' reliance on complex understoreys, usually the first vegetative layer cleared for development (Shanahan, Possingham, & Martin, 2011). With Australia's long history of co-evolution between Australian flora and invertebrates, the replacement of native understories by non-native flora results in reduced levels of insect diversity and abundance (Murray et al., 2007; White, Antos, Fitzsimons, & Palmer, 2005). Reductions in specialised invertebrates in turn, have placed resource-limitation pressure on insectivorous functional groups (McKinney, 2008). Small-bodied species have also been disproportionately impacted by habitat fragmentation. Compared to large-bodied birds, small birds are exposed to greater risk and energy costs crossing transformed areas (Shanahan et al., 2011). In addition, where there is a loss in low vegetative cover, small-bodied species, along with understorey- and ground-nesters, are more vulnerable to exclusion by larger aggressive competitors (Catterall, 2004; Kath, Maron, & Dunn, 2009). Loss of vegetative complexity also increases avian exposure to invasive mesopredators such as cats and foxes in Australia (Major, Christie, & Gowing, 2001; Olsen, 2008; Sewell & Catterall, 1998; Shanahan et al., 2011). Foxes in particular prefer the lower-levels of disturbance found in rural environments for hunting, compounding the impacts of habitat clearing in these environments (Saunders, Coman, Kinnear, & Braysher, 1995).

Whilst such insights underline the myriad impacts associated with anthropogenic activity, it remains difficult to prioritise management of these threatening processes within heterogeneous anthropogenic environments. Managers must consider controlling predators and overabundant competitors, improving landscape connectivity, mitigating pollution and restoring habitat; all with diminishing conservation funds (van Dijk, Mount, Gibbons, Vardon, & Canadell, 2014). Our study aims to prioritise management effort by identifying which functional traits are being lost across an anthropogenic landscape gradient in Greater Brisbane, Australia. We use time-series data for 182 species (see Table S1 in Supporting Information), to determine whether species and threatened functional groups are being equally affected in urban, rural and forested land. We test for changes in understorey-nesting, insectivorous, ground-nesting and small-bodied functional groups, all of which have repeatedly emerged as vulnerable in Australia (Catterall, 2004; Kath et al., 2009; Major & Parsons, 2010; Shanahan et al., 2011; Szabo, Vesk, Baxter, & Possingham, 2010).

We hypothesise that the species which exhibit the greatest decline and increase in prevalence will differ across the land use gradient (H1), and that different functional groups will decline in prevalence at each land use intensity (H2). Specifically, and based on the literature for Greater Brisbane, we predict that:

- in forested environments, where anthropogenic disturbance is lowest, there will be no detectable reduction in prevalence for any functional group (H3),
- in rural environments, where predation pressure is high (Saunders et al., 1995) there will be a reduction in the ground-nesting functional group (H4) and
- in urban environments, where there are a range of pressures and a high level of disturbance, all four functional groups will decline (H5).

Through gaining an understanding of which functional groups are being lost within each land use, we will be able to highlight where management effort should be allocated.

2. Methods

2.1. Study site

In order to test our research hypotheses, we evaluated bird assemblages across Greater Brisbane, Australia. The Greater Brisbane region has a diverse vegetative community, including eucalypt woodlands, wet and dry rainforest, melaleuca and mangrove forests. Brisbane is one of Australia's most biologically diverse State capitals (Catterall & Kingston, 1993; Catterall, Cousin, Piper, & Johnson, 2010). However, over two thirds of Brisbane's native vegetation has disappeared, partly due to sprawling suburban development (Coleman, 2016). Brisbane is now one of the fastest growing cities in Australia, with a 25% growth in human population from 2001 to 2011 (ABS, 2011). Given the regions' biological importance and extensive land transformation, the Greater Brisbane region is an ideal study area to examine the extent to which avian functional groups are being threatened by human development.

We chose to classify Greater Brisbane into three levels of land use intensity. To accurately classify the region we used landsat-image derived vegetation maps created by Lyons, Phinn, and Roelfsema (2012). These maps, covering an area of 14,600 km^2, classified South-East Queensland into 11 land cover types, are high-resolution (25–30 m^2) and have a calculated accuracy of at least 80% (Lyons et al., 2012). The entire record of maps span from 1972 to 2010. However, imagery was not available for every year. Thus, we selected the largest series of continuous annual maps, ranging from 1999 to 2008, for our land use classification.

We combined the 11 original land cover classifications into three land use intensities, to represent a gradient of urban activity (Fig. 1, Fig. S2 & Table S2). Urban (high intensity) land had a high to moderate density of human settlement, rural (moderate intensity) land had low canopy cover, but also sparse human settlement, and forested (low intensity) land had high to complete vegetative cover (see Lyons et al., 2012 in conjunction with Table S2 for further detail). Although these maps would have permitted additional land categories, it was important to maximise the number of bird lists available within each land use, to ensure robust outputs from our selected analysis method.

2.2. Avian citizen survey data

We used presence-only bird lists, available from 1999 to 2008, from the New Atlas of Australian Birds (hereafter the Atlas). The Atlas is Australia's largest and longest-running bird survey database (Barrett, Silcocks, Barry, & Cunningham, 2003). Volunteers are free to choose the location, date, time, search method and area covered by their survey, and do not use checklists. These details are included, along with the species list, record ID, observer ID and GPS survey locational accuracy, within the Atlas (Barrett et al., 2003).

The Atlas surveys are conducted using one of four methods: 2-ha area searches for 20 min, area searches within a radius of 500 m or 5 km, for at least 20 min, or incidental observations. Our study followed the methodology set out by Szabo et al. (2010) to filter records. We excluded records which did not include information on survey location, accuracy, method or area covered, or had a GPS survey locational accuracy of less than 500 m. We also excluded incidental sightings, poorly sampled species (< 10 observations or < 1 observation/year) and species lists of five or fewer species. Incidental sightings were removed because they may have introduced species' bias if observers only

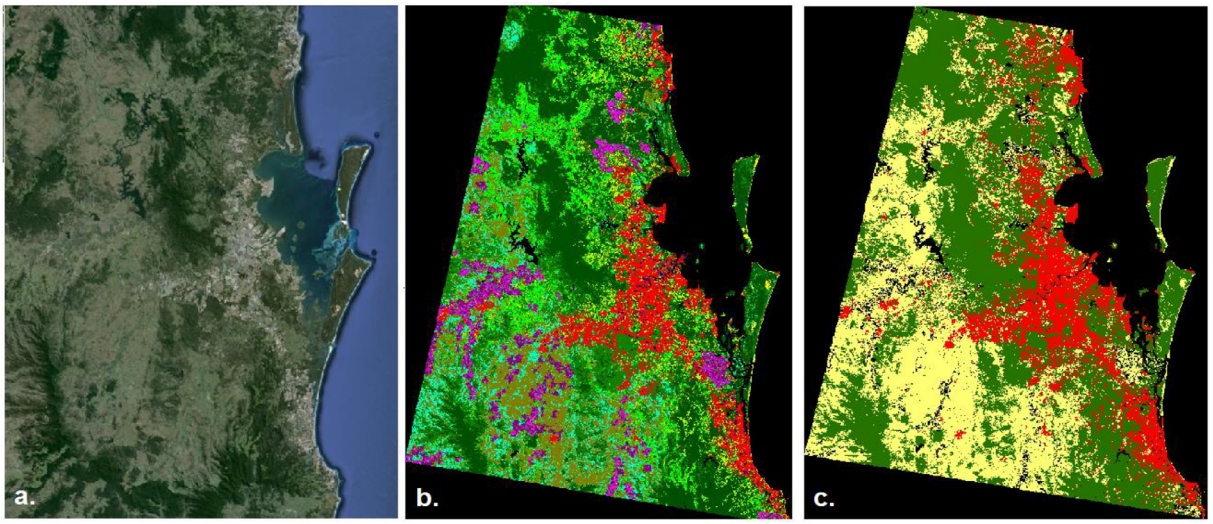

Fig. 1. Method for rezoning land cover maps a) Google Earth image over the study area (Google, 2012) b) Landsat vegetation land cover maps utilised in this study, each colour represents a different vegetation and land cover type (Lyons et al., 2012) c) Reclassified land cover map separated into 3 distinct land use intensities High (urban) = red, Medium (rural) = yellow, Low (forested = green). (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

recorded rare or interesting species (such as raptors). For lists which included species abundance information, these data were converted to presence-only lists before being included within our dataset. Migratory species, shorebirds, seabirds and nocturnal species were also excluded, as our study aimed to identify trends on the resident avian assemblage present across the Greater Brisbane region.

Following these quality checks, bird lists were plotted onto the corresponding land use maps for that year, on the geographic coordinate system WGS1984. Thus we could identify whether a bird list had been recorded in urban, rural or forested land. Where the land use changed from one year to the next, so too did the land use assigned to a bird survey at that same location. To account for a potential 500 m inaccuracy in the recorded GPS survey location, we calculated the dominant land use within a 500 m radius of each bird record, to determine in which land use each survey took place (Fig. S3) (ESRI, 2011). A total of 16,771 bird lists were plotted onto the land use maps which resulted in 6178 lists (142 species), 2110 lists (146 species) and 8483 lists (175 species) being available for analysis in urban, rural and forested land uses respectively (see Table S1 for full list of species).

For our selected analysis method, List Length Analysis (LLA), it was also important to reduce statistical noise and maximise power by optimising a) the evenness in the data (e.g. records/year, average list length/year, observer no/year and observer activity) b) the total number of records in our dataset (i.e. individual observations), and c) total number of species included in our analysis, to (Szabo et al., 2010). To meet these requirements, we examined the statistical adequacy of our dataset to ensure it did not include systematic biases through time. We found that our datasets demonstrated good evenness over time with a wide range of record lengths available in urban, rural and forested land uses and at least 142 species recorded within each land use type (Fig. S7). The range of list lengths was large, with the largest list a minimum of ten times as large as the smallest list for all data subsets. The median lengths of each bird list in each land use was even over time (Fig. S7) (Barnes, Venables, & Morris, 2012). The most active 10% of observers were generally responsible for generating 42, 47, 53, 55, 53 and 50 percent of the total number of lists available for each time period from 1998 to 2008 (Fig. S6). Therefore, whilst some observer bias exists, the bias is stable across years and should not affect the estimated change in prevalence generated for each species.

2.3. List Length Analysis

LLA allows the detection of trends in species prevalence (i.e. the proportion of lists on which the species occurs) using presence-only list data, collected with unknown effort (Barnes, Szabo, Morris, & Possingham, 2015). The approach assumes that the list length, which is the number of species on a given list, scales with detectability. Therefore, on a given sampling event, the length of a list controls for complex factors which affect detectability – including survey effort, observer skill and weather conditions. LLA models the monotonic increasing relationship between the probability of observing a particular bird species and the total number of species observed in the survey, i.e. between the probability of observation (P(obs)) and list length (L). Species that violate the assumption that the reporting rate of species is higher on shorter lists are eliminated from the analysis.

LLA uses a Bayesian logistic regression model and Markov Chain Monte Carlo (MCMC) sampling to determine the probability of a given species appearing on a given list, as a function of list length and survey year (Szabo et al., 2010) as follows:

$$\text{logit}\,(\Pr(\text{obs})) = a_1 + a_2\log(\text{L}) + a_3\text{Yr}$$

Pr(obs) gives the probability of observation for a species, L is the list length, and Yr is the year of observation centred on 2003 – the midpoint of the dataset. The coefficients $a_{1\text{-}3}$ each have minimally informative normal prior distributions (normal distributions with means equal to zero and standard deviation equal to 10,000). a_1 is the intercept term and reflects the overall prevalence of the species at the temporal midpoint (2003). a_2 describes the multiplicative effect of increasing list length, and acts as a complex proxy that controls for effort, observer skill, and conditions. a_3 is a fitted term for change in prevalence per time step (for example per year for annual data) (Szabo, Vesk, Baxter, & Possingham 2011). In this study, we are most interested in the a_3 value because this value denotes the temporal change in prevalence of each species (Szabo et al., 2010). Notably the a_3 value for each species is relative to the entire avian assemblage. Therefore, if most species experienced some decline estimates may be conservative (Szabo et al., 2010). A key assumption of this method is that detection probabilities are stable through time. This assumption is most likely to be violated for studies spanning decades for example, due to technological advances and changes in habitat structure. Given our study covers a large spatial area and just ten years, we considered that changes in

detectability would be limited (Szabo et al., 2011).

In our study, we ran LLA separately for lists in urban, rural and forested lands in R.2.15.1 (RCoreTeam, 2012) using the 'Liszt' package (Barnes & Venables, 2012) in conjunction with the Jags package for R, 'R2Jags' (Plummer, 2003). The 'Liszt' package uses MCMC resampling with 20,000 iterations and a burn in of 5000 iterations. These iteration values were based on conservative estimates of minimum iterations required to achieve chain convergence (Barnes et al., 2015). Gelman Rubin statistics ($\widehat{R}$) were used to assess convergence, with an upper threshold of $\widehat{R} = 1.1$. Species which had $\widehat{R} > 1.1$ were eliminated from further analysis.

2.4. Trait analysis

We selected insectivorous, understorey-nesting, ground-nesting and small-bodied functional traits for trait analysis, based on their prevalence in the literature as functional groups which are negatively affected by anthropogenic land uses (Catterall, 2004; Kath et al., 2009; Major & Parsons, 2010; Olsen, 2008; Shanahan et al., 2011; Szabo et al., 2010). We used the Handbook of Australian and New Zealand Birds (Higgins, 1999; Higgins & Davies, 1996; Higgins & Peter, 2002; Higgins, Peter, & Cowling, 2006; Higgins, Peter, & Steele, 2001; Marchant & Higgins, 1990, 1993) to assign these traits:

Insectivores: Species which specialise in foraging primarily on invertebrates and rarely forage on seeds, fruits or other substrates.

Understorey-nesting species: Species that nest at or below 1.5 m, but not on the ground.

Ground-nesting species: Species that nest on or below the ground (burrows).

Small-bodied species: Species which weigh less than 67 g. A threshold of 67 g was chosen as this was the median body size of the complete avian assemblage in our study.

Using the outputs of LLA we ranked species based on their estimated change in prevalence (a_3 values) and then used simple permutation tests to determine whether particular traits were significantly concentrated at increasing or decreasing trajectories. We used 1,000,000 permutations to compute the sampling distribution of each functional group's mean a_3 values to understand whether the null hypothesis, that there is no difference in the sampling distribution of each functional group's a_3 values and the sampling distribution of all species' a_3 values, was true (Breed, Stitcher, & Crone, 2012; Sham & Purcell, 2014). Species whose credible intervals a) crossed zero and b) were highly variable (97.5% credible interval range > 0.5) were not included in this additional analysis. Permutation analysis was performed separately for each land use type (urban, rural and forested). To simplify interpretation of this study's methodology we have developed Fig. S1 in Supporting Information for a step-by-step summary.

3. Results

From 1998 to 2008 urban land cover increased in area over time, while rural land area decreased, and forested land showed no overall change (Fig. S5). Species which exhibited the greatest change in prevalence differed between each category of land use (Figs. 2–4). The species which declined most in prevalence in urban, rural and forested land uses were *Acanthiza (Geobasileus) reguloides* (Buff-rumped Thornbill), *Burhinus grallarius* (Bush-stone curlew) and *Cinclosoma punctatum* (Spotted quail-thrush) respectively (Figs. 2–4). The number of species declining and increasing in prevalence (i.e. their credible intervals did not cross zero) was similar between each land use, but greatest, in both cases, in forested lands (35 species in urban, 34 species in rural and 37 species in forested were declining, while 27 species in urban, 24 species in rural and 28 species in forested land uses were increasing in prevalence). Overall there were consistently more species declining than increasing in prevalence across all land uses (Table S3).

Prior to the permutation tests, 12 species in urban land uses, four species in the rural subset and ten species in forested areas were eliminated from the analysis as their a_3 values demonstrated high variability and, for four species, MCMC chains did not converge ($\widehat{R} > 1.1$) (McCarthy, 2007; Szabo et al., 2011) (see Table S3 for full list of mean a_3 values). Permutation tests demonstrated that understorey-nesting and small-bodied species were more likely to be declining in prevalence in urban environments ($p < 0.005$, $p < 0.025$; Fig. 5). In both rural and forested land uses no significant decline in prevalence was detected for any one functional group (Fig. 5).

4. Discussion

As expected, the species which exhibited the greatest decline in prevalence differed between urban, rural and forested environments (supporting H1) (Figs. 2–5). Given species diversity is highest in forested environments and second-highest in rural land uses, one may surmise that these differences are because the species declining in rural environments have already disappeared from urban environments. However, close examination of our data demonstrates that, in Greater Brisbane, species declines are not consistent across disturbed land uses. For example, some species which have disappeared from rural environments still persist in urban and forested land uses (Table S3). Hence, it is likely that different species are being affected by distinct processes at each land use intensity, affecting their distribution. Nonetheless, there do exist some parallel trends across all land uses. In urban, rural and forested environments there was consistently a greater number of species declining in prevalence than increasing. In addition, within each land use only a handful of species were becoming more common. Such results highlight that the avian assemblage is becoming more homogeneous in disturbed environments and, perhaps more concerningly, in neighbouring forested environments where anthropogenic impacts should be minimised (Figs. 2–5).

Our analysis of individual functional groups revealed that there were differences between patterns of trait loss between land uses (supporting H2). Only in urban environments was there evidence that understorey-nesting and small-bodied species are being disproportionately affected by anthropogenic land use. As expected there was no discernible decline for any of the tested functional groups in forested environments (supporting H3). However, the lack of functional group declines in rural environments and decline in only two groups in urban environments was unexpected (refuting H4 and H5). It is possible that, in rural environments of Greater Brisbane, there were no new influential anthropogenic impacts driving discernible shifts in avian assemblages during our study period. It should be noted these rural environments were developed up to the mid-1900s and are now contracting due to urban development (Catterall et al., 2010; Coleman, 2016; Queensland Government, 1998) (Fig. S5). In contrast urban environments continue to grow and transform at a rapid rate (Coleman, 2016), meaning that more dramatic shifts in avian assemblages are detectable (Fig. S5).

Within urban environments, established findings indicate that the reduced prevalence of small-bodied and understorey-nesting species is due to the combined pressures of land clearing (Shanahan et al., 2011), avian predation (Debus, 2006), and competition (Catterall, 2004; Maron et al., 2011; Szabo et al., 2010) (Fig. 2). Interestingly, we detected an increase in *Manorina melanocephala* (Noisy Miner) prevalence only within urban environments during the study period (Table S3). Studies by Maron et al. (2011) and Clarke and Oldland (2007) demonstrate that these communal, aggressive honeyeaters can have a significant impact on avian assemblages, particularly for small-bodied birds. Our study indicates that there may be merit in investigating this interaction further, to prioritise conservation actions. Meanwhile, as small-bodied and understory-nesting functional groups are particularly dependent on understoreys for protection, revegetation efforts should

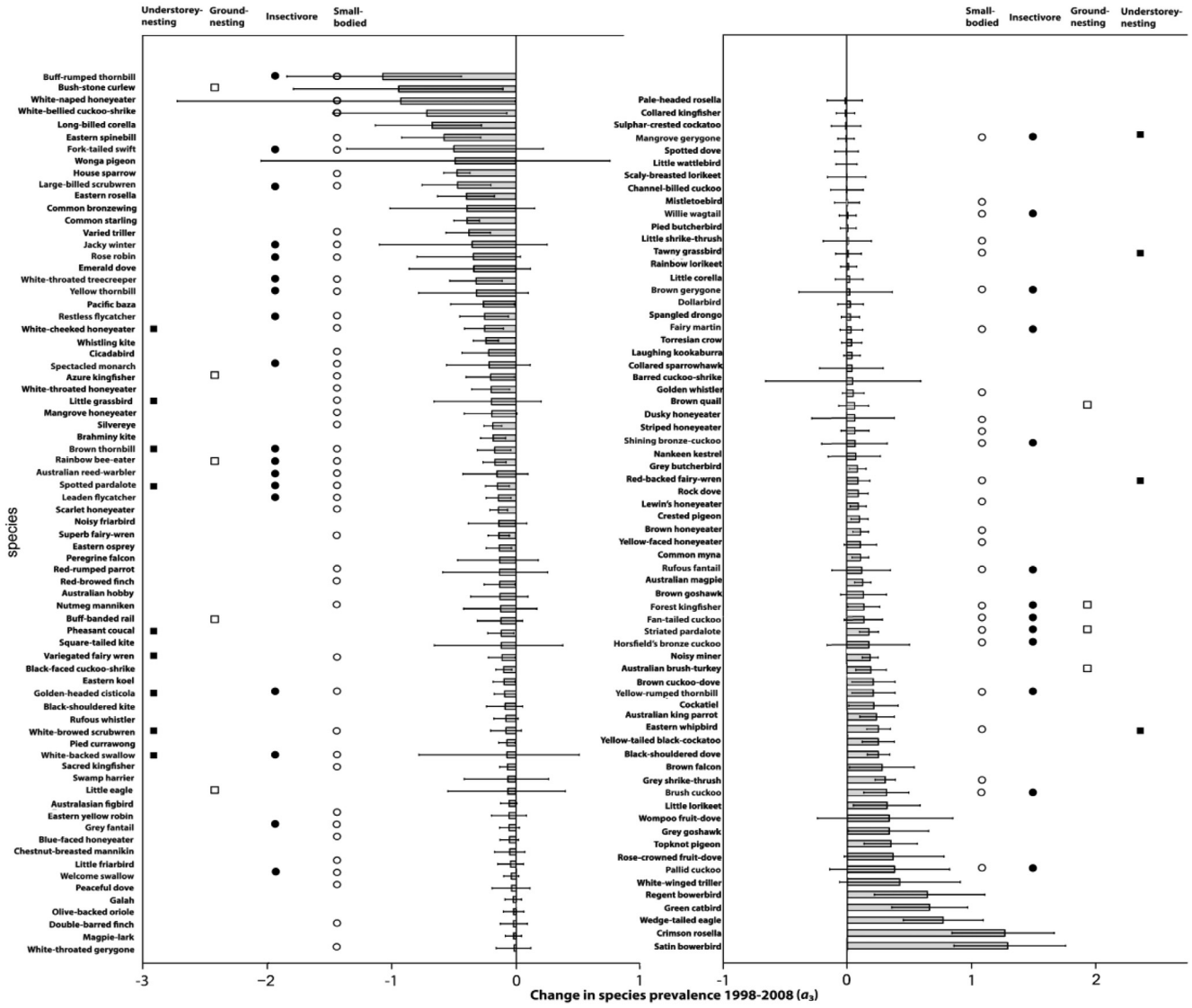

Fig. 2. Caterpillar plot demonstrating change in prevalence (a_3 values) over time in urban land uses. Each bar represents a different species. Error bars represent the 2.5% and 97.5% credible interval. A negative score indicates the species has decreased in prevalence in urban lands and a positive score indicates an increase in prevalence over time.

focus on increasing the density of understoreys in urban environments (Shanahan et al., 2011).

Our study also reflected international and national trends. These include a reduced prevalence of the typical introduced urban-adapted species, *Passer domesticus* (House Sparrow) and *Sturnus vulgaris* (Common Starling) in urban lands and increased prevalence of *Acridotheres tristis* (Common Myna) across all land uses (Fig. 2) (Maron et al., 2011; Shanahan, Strohbach, Warren, & Fuller, 2014; Shaw, Chamberlain, & Evans, 2008). Concerning declines in particular species were also highlighted by this study. For example, *Burhinus grallarius* (Bush Stone-Curlews) demonstrated the second-largest decline in urban environments and on average declined substantially in rural lands (Fig. 2). Populations of *B. grallarius* were once abundant in South-eastern Australia. However, their distribution has now drastically contracted due to land transformation and fox predation (Department of Environment, 2006). Queensland remains one of few strongholds for South-East Australian populations of *B. grallarius* (Queensland Wader Study Group, 2010) and our analysis underscores that even these populations are declining in prevalence. Such results provide compelling evidence that decision-makers should be targeting *B. grallarius* conservation. LLA also reveals trends which may be interesting to specialist research, community or government groups. For example, we identified declining trajectories for *Acanthiza reguloides* (Buff-Rumped Thornbill); *Acanthiza lineata* (Striated Thornbill) and *Acanthiza pusilla* (Brown Thornbills) across all land uses, providing guidance for prioritisation of conservation resources.

Despite these novel insights into individual species' trajectories, the outcomes of LLA must be considered along with the method's limitations (Isaac, van Strien, August, de Zeeuw, & Roy 2014; Szabo et al., 2010). While LLA is useful for identifying trends in unstructured datasets, these analyses do not replace the role of standardised data, which are critical for quantifying species declines (Szabo et al., 2010). The outcomes are always relative to the entire species assemblage and LLA treats each visit to a site as independent. Therefore, if the proportion of visits to a particular site changes over time, compared to other sites, then results could become skewed (Isaac et al., 2014). We partially alleviated this issue by splitting our dataset into land types and ensuring that the analysis of species prevalence for well-forested biodiverse environments (which may become more popular over time) are not being compared to urban landscapes (which may become less popular amongst birders). We also analysed spatial clustering of our records in ArcGIS prior to analysis and found that while records did

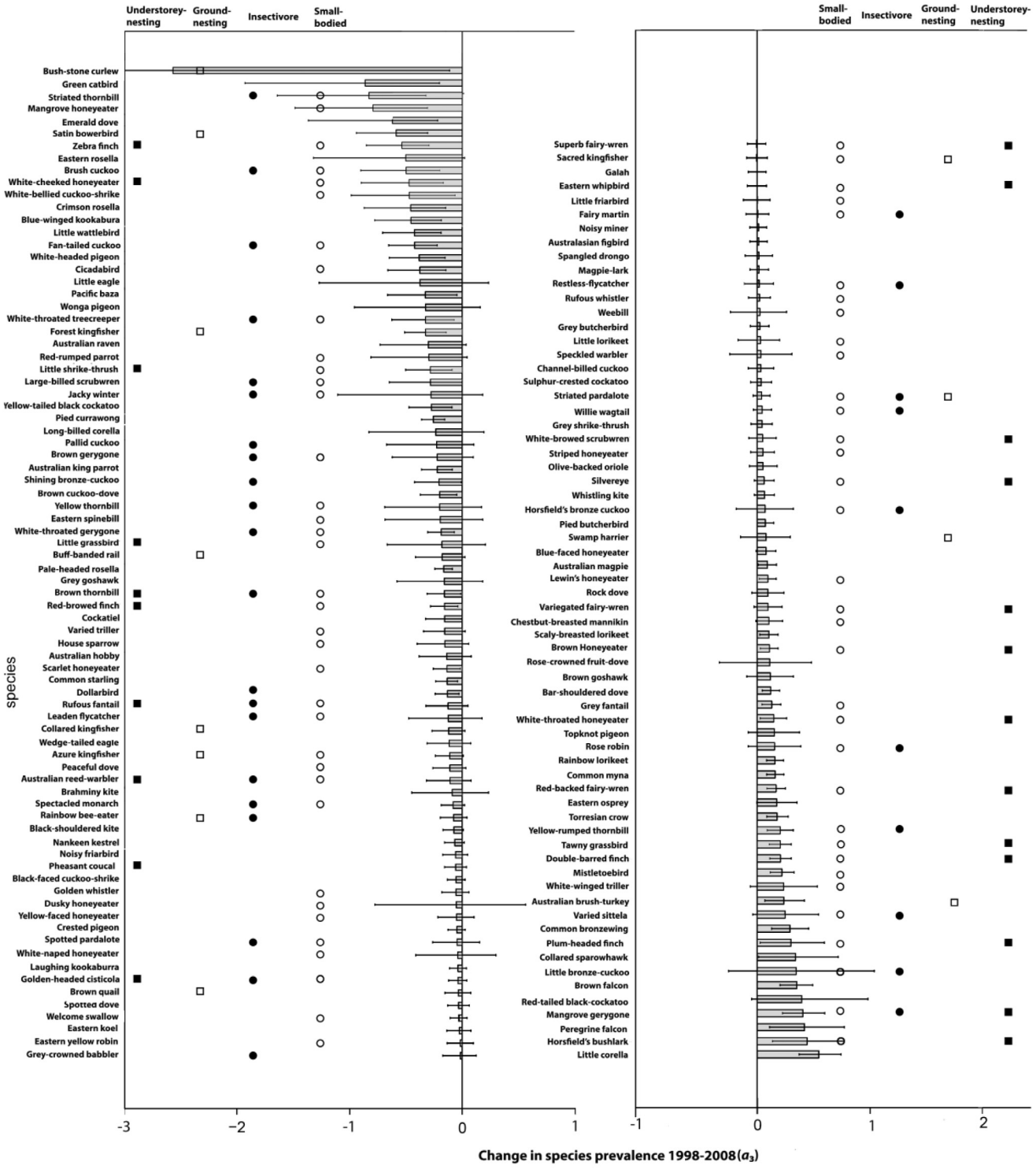

Fig. 3. Caterpillar plot demonstrating change in prevalence (a_3 values) over time in rural land uses. Each bar represents a different species. Error bars represent the 2.5% and 97.5% credible interval. A negative score indicates the species has decreased in prevalence in rural lands and a positive score indicates an increase in prevalence over time.

cluster around urban centres the degree of clustering did not change over time. When using LLA it is important to calculate these biases to ensure the data are suitable for the method.

Future studies could readily build on our method and incorporate additional multi-variate parameters within the LLA model. Such changes would ensure that interactions between functional groups across distinct land uses are better accounted for (Barnes et al., 2015). A more complete exploration of traits; including territorialism, communalism, longevity, resource-use flexibility, relative brain size, clutch-size and dispersal distance may also further explain the dynamics of avian community change (Aronson et al., 2014; Croci et al., 2008; Kark et al., 2007; Lepczyk et al., 2017; Shochat, Lerman, & Fernandez-Juricic, 2010; Szabo et al., 2010). Such studies would provide valuable insights using minimal resources, through harnessing existing datasets.

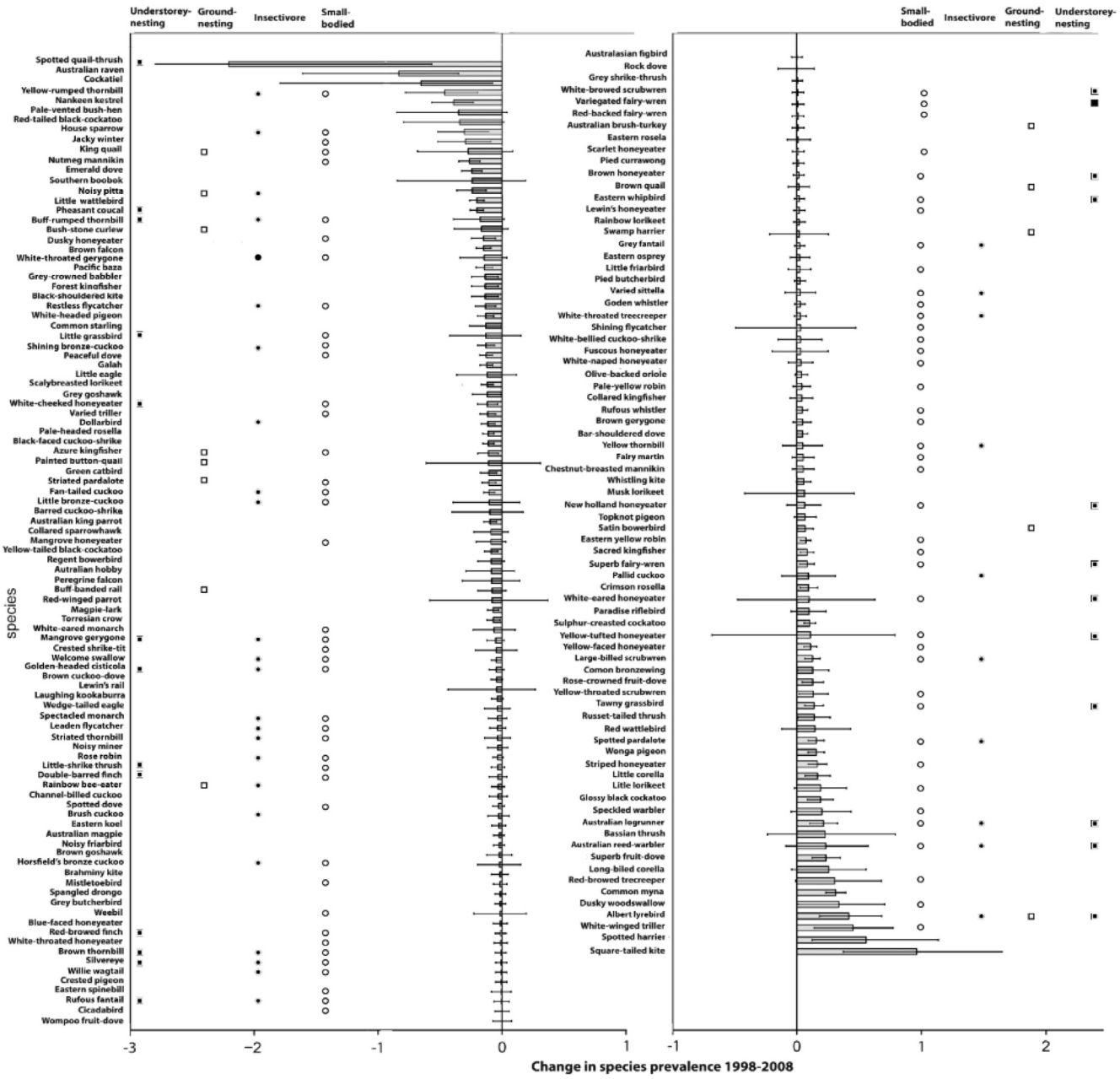

Fig. 4. Caterpillar plot demonstrating change in prevalence (a_3 values) over time in forested land uses. Each bar represents a different species. Error bars represent the 2.5% and 97.5% credible interval. A negative score indicates the species has decreased in prevalence in forested land and a positive score indicates an increase in prevalence over time.

5. Conclusions

Our analyses highlighted that the species declining in prevalence differ across the land-use intensity gradient, underscoring the importance of tailoring management efforts to mitigate key land use threats. In urban environments of Greater Brisbane we found significant declines in understorey-nesting and small-bodied functional groups, highlighting that planners should preserve dense protective understories. We also detected an increase in the abundance of the aggressive honeyeater *Manorina melanocephala* in urban environments, indicating that the influence of aggressive avian competitors should be investigated. Our results flagged that some species are declining across all land uses, providing guidance and further support for expanding conservation programs in the region. While LLA should not be used alone to drive conservation strategies, this simple and inexpensive approach represents a useful first step in identifying changes in biodiversity (Isaac et al., 2014; Szabo et al., 2010). Unfortunately, investment in monitoring is typically low in Australia (van Dijk et al., 2014) making it difficult to allocate funds. LLA can be applied to variable, unstructured datasets to highlight which species are at risk, prioritise conservation actions, communicate the importance of urban biodiversity conservation and target monitoring effort. Such insights can assist in maintaining avian diversity and preserving human interactions with nature, within the growing anthropogenic matrix (Fuller, Irvine, Devine-Wright, Warren, & Gaston, 2007; McKinney, 2008).

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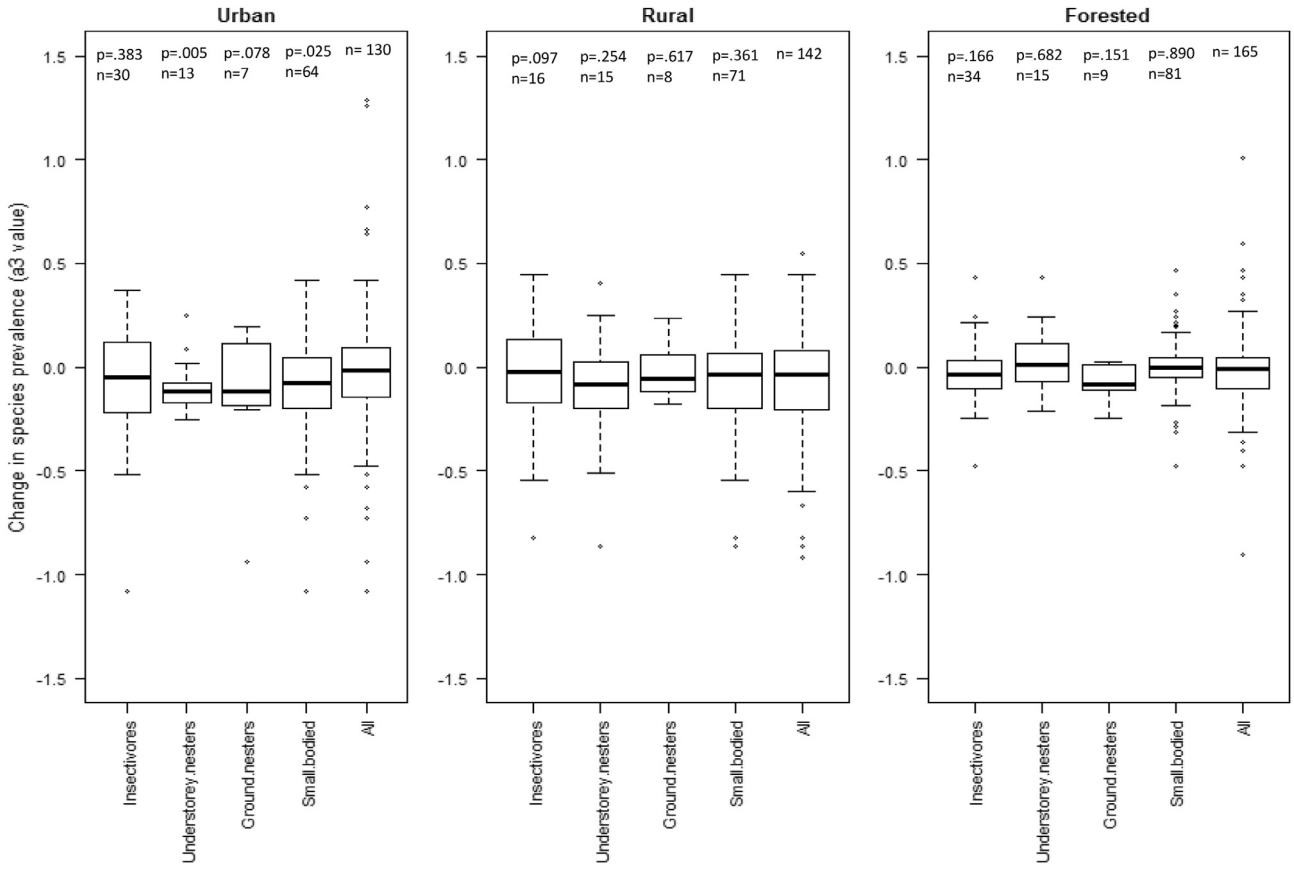

Fig. 5. Species' mean change in prevalence (a_3 values) within urban, rural and forested land uses for: Insectivores, Understorey-nesters, Ground-nesters, Small-bodied species and all species recorded. P values were determined using permutation analysis.

maps. We would also like to acknowledge Birds Australia and the volunteers responsible for collecting the extensive records that make up the publicly available Birds Atlas database. Finally, we are grateful for the valuable comments and suggestions provided by the anonymous reviewers and the associated editor of the Journal of Landscape and Urban Planning, Dr. Christopher A. Lepczyk.

Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at https://doi.org/10.1016/j.landurbplan.2018.07.006.

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